Chapter Three: Issues Affecting Biodiversity

The biodiversity of an area is nothing more or less than the sum of all the species that inhabit that area. Each species has its own set of environmental requirements and a certain ability to interact with other species in an area that allows the population of that species to continue to exist over time. Why certain species occupy one area and not another depends on many factors and on the interactions of these factors. The major factors determining the biodiversity of New England are glaciation, the range of climate from the coast to the western hills, the geology of the area (largely a function of past glacial occurrences), and the resulting diversity of habitat types found in Massachusetts.

Glaciers have periodically covered New England (including Massachusetts) and then receded. This has occurred many times throughout history. The landscape of broad highlands, narrow valleys, and north- to south-running hills that we see today, along with areas of exposed bedrock and deep deposits of sand and gravel, are a result of the last glacial period some 10,000 years ago. During the last glacial period, sea level decreased to the point where the present-day islands of Nantucket and Martha's Vineyard were connected to the mainland. As the glaciers receded, some species were able to re-colonize the area. With no inland access to the Ohio and Mississippi drainages or to points east of the Hudson River, the only route left for freshwater fish to re-inhabit the area was to move north along the coastline. Many species which could not tolerate the salt and brackish conditions there were unable to move northward to repopulate the area, leaving the fish fauna of Massachusetts much less diverse than, for example, abutting New York. Glaciers also created a large number of small and large depressions where buried chunks of ice melted. Some of these depressions fill with water on a seasonal basis, creating important vernal pool habitat for amphibians. Larger ones that intersect the water table on a permanent basis are called "kettlehole ponds" and have unique environmental characteristics typically involving low nutrient levels and fluctuating water levels.

Climate plays a significant role in determining habitat. The climate has certainly changed in this area over time from the frozen period of the last glacial epoch to the much more temperate conditions we see today. The highest recorded temperature in Massachusetts in recent times was 107° F and the lowest was –35 ° F. Average annual precipitation ranges from about 40 inches in the Connecticut River valley to about 50 inches in the higher altitudes of the Berkshire Hills. Precipitation in coastal areas averages about 45 inches annually (National Water Summary). Widespread flooding caused by intense rainfall combined with warm temperatures and snowmelt, and occasional "northeaster" and tropical storms, create and maintain floodplain habitats.   New England in general, and Massachusetts in particular, is occasionally hit by devastating hurricanes. Periods of below normal precipitation with resultant droughts increase the likelihood of naturally occurring fires. Today these fires are usually brought under control quickly, which allows habitats once maintained by these disturbances to degrade.

The direction and speed at which our climate is changing is the focus of a great deal of research these days. These changes will favor some species and negatively impact others. The review process for the CWCS is to take place every five years. As these and other impacts to the environment change over time the list of Species in Greatest Need of Conservation will likely have to be amended. To address the issue of global climate change and because the evidence points to increased emissions of Greenhouse Gasses (GHG) as the cause for increases in global temperatures, the Office of Commonwealth Development published a document titled *Commonwealth of Massachusetts Climate Protection Plan Spring 2004* to address ways we all can reduce the amounts of GHG emissions from within the Commonwealth. Actions called for in the plan include, "The plan focuses on a range of strategies to achieve significant near-term reduction of GHG emissions." These strategies give priority to pollution reductions that are compatible with economic growth measures which ease the transition to cleaner and less expensive energy resources, and which retain a higher proportion of the states energy dollar within Massachusetts. These strategies encourage public agencies, businesses, industries, and citizens to take cost effective, common sense steps toward reducing GHG emissions in ways that also advance other important state priorities and objectives.

Habitat Types

Massachusetts falls within two ecoregions of the United States: the **Northeastern Highlands** and the **Northeastern Coastal Zone.** These are areas of relatively homogeneous ecological systems, including vegetation, soils, climate, geology, and patterns of human use. These two ecoregions have been further divided into thirteen sub-ecoregions as defined by Griffith et al. (1994) (Figure 2). Massachusetts lies at the southern edge of forest types more typical of Maine and the eastern Canadian provinces. Spruce-fir-northern hardwoods and northern hardwoods-hemlock-white pine exist in the higher elevations of western Massachusetts. The state also lies at the northern edge of forest types found along the mid-Atlantic; thus central hardwoods-hemlock-white-pine and pitch pine-oak can be found throughout Cape Cod and eastern and southern portions of the state. Transitional hardwoods-white-pine-hemlock forests are found throughout the majority of the remainder of the state.

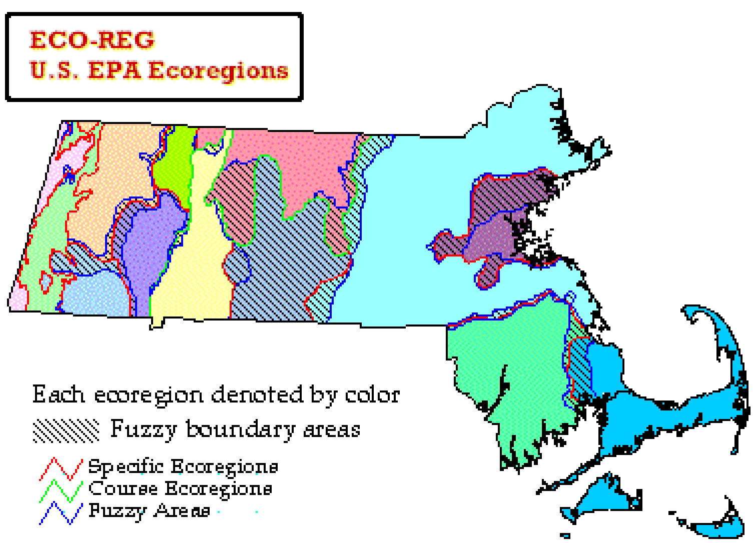

Figure 2: USEPA Ecoregions of Massachusetts

A. Habitat Destruction by Development

By far the greatest contributor to the loss of species and habitat diversity in Massachusetts has been the destruction and fragmentation of habitat by residential, commercial, and industrial development. About one-quarter of the land area of Massachusetts is now developed – about 1.2 million acres of the state's total of just over 5 million acres, as demonstrated by interpretation of aerial photographs taken in 1999. A comparable analysis of the landscape of the Commonwealth in 1971 showed only 17% (or about 857,000 acres) was developed at that time.

Low-lying lands along the coast and river valleys are areas where, historically and currently, development for human use has been mostly highly concentrated. These are also areas, such as Cape Cod, the Islands and the Housatonic and Connecticut River Valleys, that contain important components of the state's biological diversity. Over the years conservationists in the state have protected many of the mountain tops, which do, of course, support other components of our biodiversity, but much of the lowlands where many species live are relatively heavily developed, with only fragmented open space remaining. In between the mountain tops and the river valleys and coastal lowlands are rolling areas under increasing development pressure.

In recent decades, the loss of habitat to development has been compounded by ever-greater acreage used for each residential unit: *from 1950 to 2000, the population of Massachusetts increased by 28%, but the area of developed land has increased by 200%* (NHESP, 2001; Breunig, 2003).

For animals, habitat loss comes in several ways. Direct destruction in which the habitat is entirely eradicated is one widespread extreme, but small physical losses, which may not seem particularly drastic individually (e.g., houses built one-by-one in an expanse of undeveloped land such as coastal heathland), can also, collectively, produce very significant losses of habitat. While the landscape may remain fairly natural looking, the habitat is disrupted and sources of disturbance, such as noise, lights at night and exotic species, are introduced. New species that are adapted to disturbance come into the environment and change the habitat of the native species by adding competitors or predators or by causing structural changes to the ecosystems (for example, creating more, or less, understory in a forest, or different tree heights or types of trees). Some native species are also subject to increased stress by the presence of people. Exploring or predaceous pets can adversely impact the nesting success or survival of native ground-nesting or -feeding birds. This effect is particularly strong in coastal heathland and grassland communities – and in interior forests.

In general, using an ecoregion, or other large area basis, for tracking types and rates of land-use change can give an indication of the degree to which native biodiversity is threatened in the larger region. Development threats to biodiversity can be effectively assessed by tracking the actual amount of land use change using a constant measure such as acreage. Data are available to complete such analyses, including land use and housing start figures and the number of acres in open space, such as Massachusetts Audubon did in their report, *Losing Ground*. These data are used to determine the rate of land conversion and the actual acreages of each land-use classification. In *Our Irreplaceable Heritage,* the Natural Heritage & Endangered Species Program included a comparison between development rates in two ecoregions in Massachusetts, the Connecticut River Valley and the Worcester Plateau, showing that the Connecticut River

Valley ecoregion is under greater threat because a larger proportion of the ecoregion has already been developed, it has fewer acres of protected open space, and during a recent 14-year period, there were about twice as many acres developed in the Valley as the Plateau (Barbour et al. 1998). Examining maps of the different ecoregions also demonstrated that following development patterns by ecoregion rather than political boundaries can be the most effective way to analyze threats: development often follows geology and topography (easiest areas to develop first), which is not clear in looking at maps following political boundaries. Knowing and using such patterns assists in planning for land and biodiversity protection efforts.

Development threats to biodiversity and development threats to undeveloped land are not necessarily the same thing. Analysis of loss of land to development needs to be considered on a large scale, such as an ecoregion, in order to best prioritize land acquisitions to protect biodiversity. Locally each new development of land is a loss of open space, and can create reactive land acquisition to deal with imminent threats. Planning ahead based on information about biodiversity distribution or as part of a comprehensive plan such as NHESP's BioMap helps greatly in achieving a broad effort to protect biodiversity.

In using biodiversity and habitats for planning land protection, ecological boundaries should be drawn to include not only a population's immediate habitat requirements, but also enough area to allow for natural dispersal patterns, habitat buffers, and /or watershed protection that will help ensure long term viability of the population. With the amount of land development currently occurring in Massachusetts, existing and known housing subdivisions or roads can be obstacles to defining and protecting ecological boundaries for rare species or natural community occurrences in need of protection. When such obstacles occur, practical boundaries are poor representations of the ideal extent needed for habitat and biodiversity protection. Some species may not be viable in the resulting areas whether they are protected or not.

Fragmentation of ecosystems-- the breaking of large blocks of land into smaller, more isolated pieces-- disrupts the habitat for the constituent species. Fragmentation may also isolate populations of a species, leading to lowered viability. Populations of many species are able to rise and fall in different areas, moving back and forth with recolonizations after local extinctions, when those populations can be connected. However, when connections are broken, overall populations of a species may decline if the organism has difficulty in recolonizing areas when local populations die out. Another problem with the fragmentation of ecosystems is the reduction of natural community interiors. Edge-dwelling predators such as striped skunks (*Mephitis mephitis*), raccoons (*Procyon lotor*), coyotes (*Canis latrans*) and domestic cats and dogs can follow powerlines or roads into forest areas and prey upon interior-dwelling species that evolved without defenses against some of these predators. Generally, the management of small fragmented natural communities is more difficult than managing larger areas. Due to surrounding land uses, water regimes may become difficult to protect or control. Fire, as a management tool, also becomes much more difficult to use on small properties or with close neighbors. Wind has a disproportionally larger effect on edge trees than on more protected interior-growing trees.

Land use change from a wild to a developed state clearly has unintended consequences for biological diversity and ecosystem functioning and services. The intended consequences of development are to have more places for humans to live and work, better access to those places, and to provide for human recreation. Balancing the trade-offs between satisfying immediate human needs and maintaining other ecosystem functions and biodiversity, requires solid information about ecosystem responses to different land uses. Greater knowledge of wildlife and ecological aspects of development can provide a basis for assessing the trade-offs (DeFries et al. 2004). The research that may become possible with CWCS support would greatly assist us in improving such biological knowledge and improving land protection decisions.

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Losing Ground: At What Cost?

A report by the Massachusetts Audubon Society, 2003, excerpted below.

Low density, large lot residential development continues to consume forest and agricultural land in ecologically sensitive areas, according to a new Mass. Audubon report, *Losing Ground: At What Cost?*, the latest edition in its Losing Ground series. The report is based on research into changes in land use and their impact on habitat, biodiversity, and ecosystem services in Massachusetts.

While the state has seen little or no growth in single-family housing starts, residential development represents a growing proportion of land consumption. The average living area for new homes increased 44 percent between 1970 and 2002, while average lot sizes increased 47 percent in the same period. Average lot sizes more than doubled in Plymouth, Bristol, Essex, Franklin, and Hampshire counties. Particularly inefficient land consumption involving a large number of acres per new housing unit or new permanent resident could be seen in a "sprawl frontier" running through Worcester County and north of the Cape Cod Canal.

Visible development as reflected in land use data tells only part of the story, however. When parcel boundaries are considered, the true impact of development—including road building and fragmentation—is closer to 78 acres per day. The report also measures the economic impact of habitat loss, and includes the first statewide attempt to measure the economic value of "ecosystem services" provided by undeveloped land–such as climate control, water filtration, and flood control. It also calls upon citizens in the Commonwealth to work with their state and local representatives to address the problems of sprawl and habitat loss.

Specific findings of the report, which drew upon thirty years of land use and open space data and tax assessor records, include the following.

- Over 202,000 acres, or **40 acres per day**, were visibly converted to new development statewide between 1985 and 1999, equal to the entire land area bounded by Routes 128 and 95, north to Lynn and south to Quincy. Thirty-one acres of forest, 7 acres of agricultural land, and 2 acres of open space were developed each day during the period.
- Nearly **nine of every ten acres lost went to residential development**, with 65 percent used for low-density, large-lot construction
- When the total acreage of lots with new construction in the period was considered, the true impact of development was closer to **78 acres per day**. This "hidden" development impact, including road building, fragmentation, and effect of runoff, pets and invasive species, is not reflected in land use data based on aerial photography.
- While progress has been made in land protection in the recent past, 71 percent of the state's wildlife habitat – defined as forest, wetlands, lakes, ponds, rivers, streams, and open land with habitat value – **lacks permanent protection** and is at risk of development.
- Of the land area of the state delineated as the minimum area needed to protect viable populations of **rare terrestrial species, 61 percent lacks permanent protection** and is at risk of development. Because delineation of rare species habitat carries no regulatory protection, many of these "core habitats" are subject to ongoing destruction, fragmentation, and encroachment by development. Only 23 percent of the riparian land area near aquatic rare species habitat is permanently protected.

B. Fragmentation by Development

The process through which continuous forest is broken into forest patches of varying size, isolated from each other by tracts of non-forest land, is called *fragmentation* (Hunter 1996, Haila 1999). Historically, agriculture has been the most important factor driving fragmentation (Haila 1999), but in recent times, development of suburban landscapes has accounted for the bulk of fragmentation in Massachusetts (Bruenig 2003). Similarly, rivers are broken into 'patches' that are isolated from each other by dams.

In general, small fragments have fewer species than large fragments, and more isolated fragments have fewer species than less isolated fragments (Hunter 1996). Larger fragments typically have a greater variety of environments than small fragments, and each environment provides niches for species that would otherwise be absent (Hunter 1996). Also, large fragments are likely to have both common and uncommon species, whereas small fragments are more likely to have common, rather than uncommon species (Hunter 1996).

This can be true for a couple of reasons. One is that some bird species tend to avoid patches of habitat that do not greatly exceed their home range area requirements (see Robbins et al. 1989). Species that do not occur in small patches of habitat are called area-sensitive species (Hunter 1996). Another reason is that uncommon species that are not area-sensitive are less likely to occur in small patches by chance alone (Hunter 1996). For example, a species that occurs at a density of one individual per 1,000 acres across a continuously forested landscape has only a 1:100 chance of occurring in any 10-acre fragment.

Fragmentation interrupts the flow and exchange of energy and matter through aquatic and terrestrial habitats. Fragmentation threatens biodiversity by disrupting biological processes through reduction of total area of terrestrial habitat, by increasing the isolation of terrestrial and aquatic fragments from each other, and through disruption of fragments by influences from surrounding non-forest land (e.g., nest predators in terrestrial systems, flowage restrictions in aquatic systems, and invasive plants in both terrestrial and aquatic systems) (see Harris 1984, Wilcove et al. 1986, Hunter 1990, and Noss and Scuti 1994). Isolated populations in habitat fragments can suffer elevated extinction rates and loss of rare species (Forman and Collinge 1996).

Terrestrial Impacts of Fragmentation

Impacts of fragmentation on wildlife vary by species. For example, while numerous studies of breeding bird communities have documented declines in species richness in smaller vs. larger forest fragments (see Rappole 1996), some bird species appear to utilize several patches of forest as functionally continuous habitat (Haila 1999). Similarly, mortality of New England cottontails increases in small habitat fragments (<2.5 ha) compared with larger ones (>5 ha) (Barbour and Litvaitis 1993, Oehler and Litvaitis 1996), and some insects may not readily cross non-forest areas that separate forest patches (Haila 1999). As a small fragment becomes isolated from other fragments, it becomes increasingly inefficient for even highly mobile animal species to occupy it (Hunter 1996).

It appears likely that the total effect of fragmentation is non-linear relative to area – that is, the effect is negligible when fragmentation is minimal across a continuously forested landscape, but becomes very important after a certain threshold of fragmentation is reached (Haila 1999). The threshold level will vary among different species of wildlife, but overall, the connectivity of landscape pattern drops abruptly when about half of the forest area is removed (Haila 1999). Impacts of fragmentation on different wildlife species vary because a variety of causal mechanisms are often involved. For example, some bird species may suffer extensive mortality in small forest patches because predators that use adjacent non-forest areas destroy eggs (e.g., raccoons), kill adult birds directly (e.g., house cats), or both (e.g., fox).

Prior to European settlement, forest cover may have approached five million acres across what is now Massachusetts. Today, there are approximately three million acres of forest in the state (Alerich 2000) (Table 1). Forest cover varies greatly across the state, from a high of over 80% in the Berkshire ecoregions to a low of less than 20% in the Boston area (Table 2).

The loss of about two million acres of forested habitat across Massachusetts has certainly had a negative impact on wildlife, but the fact that remaining forestlands are often broken into patches isolated from each other increases the negative impacts on wildlife far beyond what might be expected if all remaining forestlands were contiguous. A measure of the degree of forest fragmentation in Massachusetts can be made by estimating the amount of remaining forest that has been isolated by fragmenting features such as roads and developments.

Bell and Scanlon (in prep) used buffering distances of 100, 300, and 1,000 m for development features with increasing fragmentation impacts (e.g., town roads, state highways, and interstate highways, respectively), and found that while >57% (about three million acres) of Massachusetts is forested today, <12% (about 600,000 acres) is buffered from fragmentation (Table 2 and Figure 3). It is sobering to note that, even within the six Massachusetts ecoregions that are still >70% forested, the amount of forest cover buffered from fragmentation ranges from 15.6% - 52.5% (Table 2). Within the four ecoregions that are currently 50-70% forested, the amount of forest cover buffered from fragmentation ranges from 4.2% - 10.4% (Table 2). These figures indicate that even the most heavily forested portions of Massachusetts have been impacted by fragmentation.

Table 1. Forest and estimated interior forest summary for Massachusetts.

Land Type	Acres	Percent of all Land	Number of Polygons	Polygon Acres		
				Average	Median	90th Percentile
All Land	5,179,350	100.0%	-	-	-	-
Forest	2,964,336	57.2%	19,701	150.5	5.2	62.7
Interior Forest	599,619	11.6%	5,213	115.1	8.5	263.7

Table 2. Forest and estimated interior forest acreage summary for Massachusetts ecoregions.

Ecoregion (Land Type Association)	**Acres**			**% of All Land**	
	All land	**Forest**	**Interior Forest**	**Forest**	**Interior Forest**
Berkshire Transition Association of the Hudson Highlands	229,616	194,201	71,475	84.6	31.1
Berkshire-Vermont Upland	433,948	374,332	157,096	86.3	36.2
Boston Basin	204,388	37,122	2,221	18.2	1.1
Cape Cod Coastal Lowland and Islands	517,667	229,608	16,312	44.4	3.2
Connecticut River Valley	339,598	142,670	12,013	42.0	3.5
Gulf of Maine Coastal Lowland	186,764	79,818	6,848	42.7	3.7
Gulf of Maine Coastal Plain	1,024,308	447,445	26,256	43.7	2.6
Lower Worcester Plateau	681,633	484,626	106,376	71.1	15.6
Narragansett-Bristol Lowland and Islands	586,635	297,700	24,676	50.7	4.2
Southeast New England Coastal Hills and Plain	233,905	136,321	15,546	58.3	6.6
Southern Green Mountains	20,500	18,775	10,690	91.6	52.1
Southern Vermont Piedmont	138,574	107,147	27,173	77.3	19.6
Taconic Highlands Association of the Taconic Mountains	81,519	72,650	42,773	89.1	52.5
Western New England Marble Valley Association of the Hudson Highlands	75,304	45,618	7,807	60.6	10.4
Western New England Marble Valley Association of the Taconic Mountains	154,549	83,745	15,346	54.2	9.9
Worcester-Monadnock Plateau	270,439	212,556	57,011	78.6	21.1
TOTALS	**5,179,350**	**2,964,336**	**599,619**	**57.2**	**11.6**

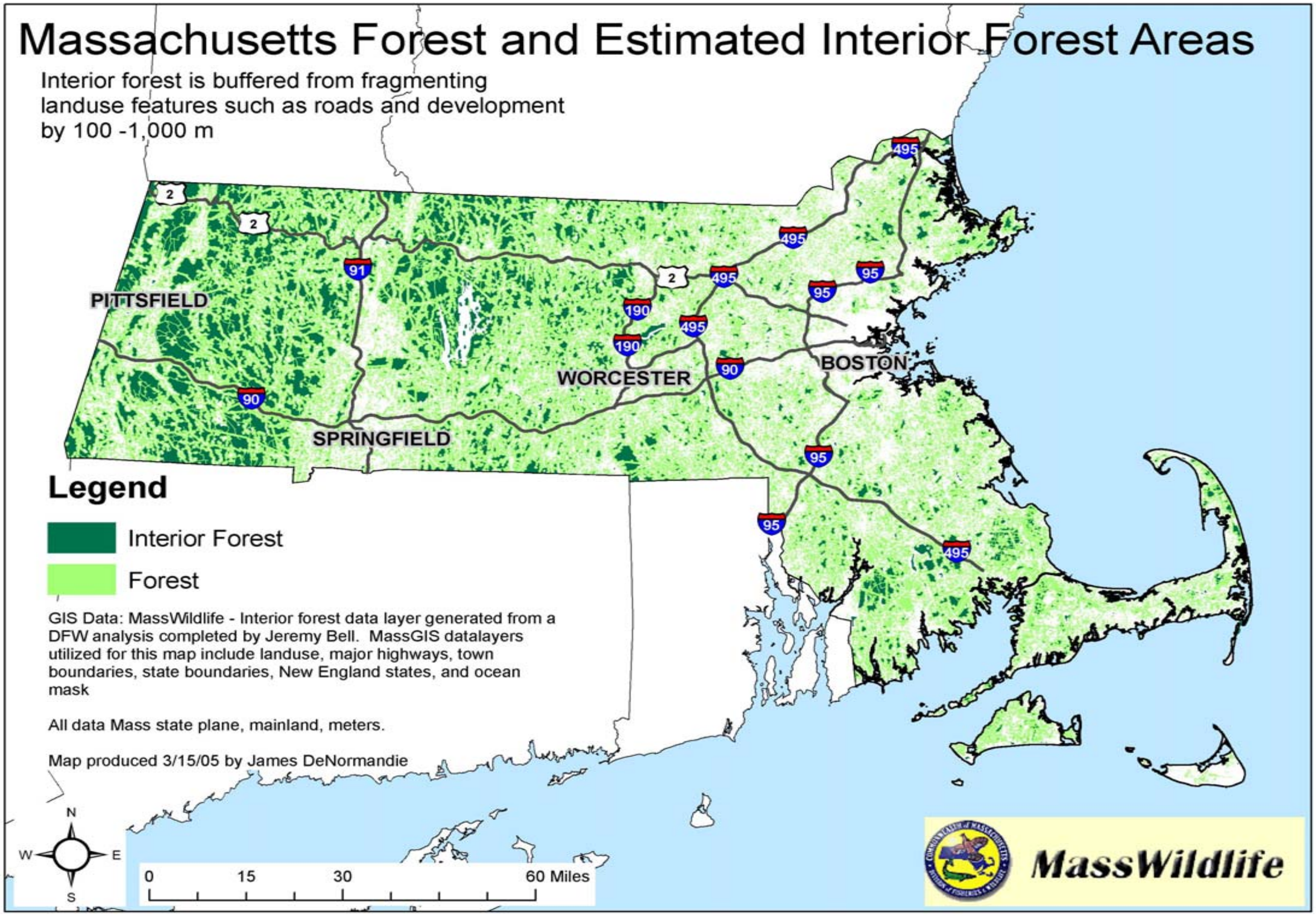

Figure 3: Forest and estimated interior forest areas.

The approximately three million acres of forest cover can be divided into nearly 20,000 polygons (Table 3), with longer distances separating forest polygons in eastern Massachusetts (88.9-106.2 m) and in the Connecticut River valley (94.5 m) and the Housatonic and Hoosic River valleys (100.3-102.5 m); and generally shorter distances separating forest polygons in central Massachusetts (84.3-89.1 m) and western Massachusetts (68.0-103.3 m) outside the major river valleys (Table 3). Similarly, un-fragmented (interior) forest can be broken into more than 5,000 polygons with separation distances of 484.9-906.7 m in eastern Massachusetts, 960.9 m in the Connecticut River valley, and 617.0-635.5 m in the Housatonic and Hoosic River valleys, vs. 456.2-474.8 m in central Massachusetts and 300.3-486.0 m in western Massachusetts outside the major river valleys (Table 3).

It is important to note that remaining forestlands do not appear to be fragmented by forest cutting activities, at least as far as breeding birds are concerned. A general pattern appears to be that predation on bird nests increases at the edge of forest fragments, but this does not happen within forested areas that contain ephemeral, internal edges that result from forest cutting practices. Specifically, no increases in nest predation rates were found in clearcut stands of northern hardwood compared to older stands (DeGraff and Angelstam 1993), and no cumulative differences in bird species richness was found across a variety of temporary forest edges between seedling, sapling-pole, large-pole, and sawtimber stands (DeGraaf 1992). Likewise, no elevation in nest predation rates were found in managed (cut) northern hardwood forests when compared to extensive, uncut forest reserves (DeGraaf 1995). These results indicate that if land remains in forest use, harvesting of renewable wood products that can support sustainable local economies will not fragment forested habitats.

Table 3. Average distance between forest and estimated interior forest polygons for Massachusetts ecoregions.

	Avg. distance (m) between polygons*	
Ecoregion (Land Type Association)	**Forest**	**Interior forest**
Berkshire Transition Association of the Hudson Highlands	84.4 ± 2.9	370.1 ± 17.1
Berkshire-Vermont Upland	85.9 ± 2.8	300.3 ± 9.8
Boston Basin	137.6 ± 4.1	906.7 ± 155.1
Cape Cod Coastal Lowland and Islands	101.0 ± 2.7	490.5 ± 33.8
Connecticut River Valley	94.5 ± 1.3	960.9 ± 77.6
Gulf of Maine Coastal Lowland	106.2 ± 2.7	484.9 ± 39.7
Gulf of Maine Coastal Plain	98.2 ± 0.7	938.9 ± 37.1
Lower Worcester Plateau	89.1 ± 1.7	474.8 ± 14.0
Narragansett-Bristol Lowland and Islands	93.1 ± 1.3	683.6 ± 31.4
Southeast New England Coastal Hills and Plain	88.9 ± 1.3	702.4 ± 34.5
Southern Green Mountains	68.0 ± 2.2	486.0 ± 71.2
Southern Vermont Piedmont	80.2 ± 2.2	431.7 ± 23.4
Taconic Highlands Association of the Taconic Mountains	103.3 ± 8.1	456.6 ± 54.9
Western New England Marble Valley Assoc. of the Hudson Highlands	102.5 ± 4.0	635.5 ± 59.3

Ecoregion (Land Type Association)	Avg. distance (m) between polygons*	
	Forest	Interior forest
Western New England Marble Valley Assoc. of the Taconic Mtns.	100.3 ± 2.5	617.0 ± 48.3
Worcester-Monadnock Plateau	84.3 ± 2.3	456.2 ± 17.0

*The minimum separation distance between interior forest polygons = 200 m. There is no minimum separation distance between forest polygons.

Aquatic Impacts of Fragmentation

Watersheds in Massachusetts have been altered by human activities for nearly four centuries. The earliest impacts were caused by agricultural and industrial expansion, later impacts were caused by massive timber harvest, and more recently by damming and industrial or urban waste disposal (Hartel et al. 2002). The Massachusetts Riverways Program estimates that there are over 3,000 dams in Massachusetts (see Figure 4). In recent decades, urban sprawl has also been a factor that has lead to substantial loss of habitat.

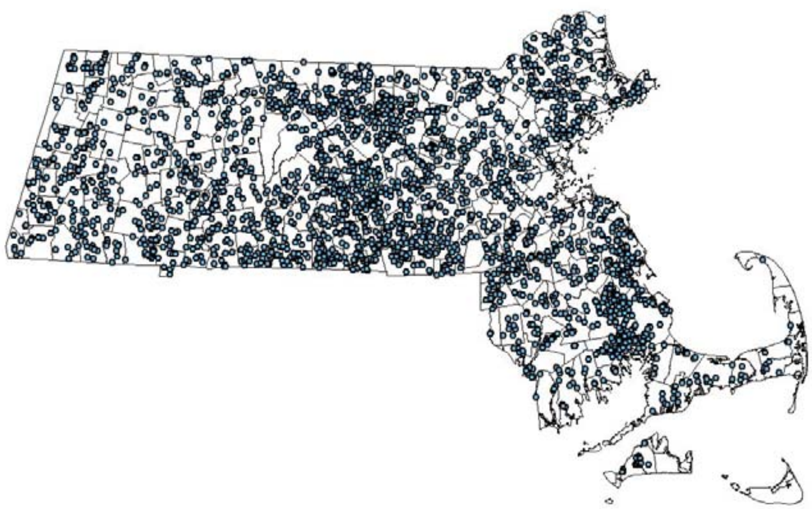

Figure 4: Map of Massachusetts depicting the location of approximately 3000 dams.

Fragmentation in streams and rivers is not a simple up and downstream issue. Streams, for example, are connected up and downstream (longitudinal), to their floodplains and floodplains to uplands (lateral), through subsurface flows, to their stream banks (vertical) and through time (temporal). Disruption of any of these parameters will lead to a degradation in the structure and function of the watershed (Williams et al. 1997).

Nationally, sources of aquatic habitat degradation include impoundment, channelization, water withdrawal, and sedimentation (Waters 1995, Instream Flow Council, in press). All of these impacts can result in fragmentation. These impacts, added to natural environmental fluctuations, cause stress to fish communities (Fausch et al. 1990). The alteration of river flow regimes associated with dam operations has been identified as one of the three leading causes, along with non-point source pollution and invasive species, of the imperilment of aquatic animals. All four of the Infrastructure Variables (density of roads, point source discharges, dams, toxic release inventory sites) used by Coles et al. (2004) to describe the urban index and its impacts coastal streams are sources of fragmentation.

Natural freshwater ecosystems are strongly influenced by specific facets of natural hydrological variability (Richter et al., 2003). Researchers have identified five critical components of the flow regime that regulate ecological processes in river ecosystems: the magnitude, frequency, duration, timing, and rate of change of hydrologic conditions. Fragmentation affects each of these components, and these components influence ecological integrity, both directly and indirectly, through their effects on other primary regulators of habitat integrity. Modification of flow thus has cascading effects on the ecological integrity of rivers (Poff et al. 1997).

Mitigating Impacts of Fragmentation

In order to mitigate the impact of fragmentation on wildlife, conversion of forestland to non-forest use must be avoided. The highest priority sites for conservation of forestland may be in the Taconic Mountains and Southern Green Mountain areas of Massachusetts where >50% of the landscape still occurs in relatively un-fragmented forest. A secondary priority may be the Berkshire Transition and Berkshire-Vermont areas of Massachusetts where >30% of the landscape still occurs as relatively un-fragmented forest.

Traditionally, the most common ways to retain land in forest use has been for government or private conservation groups to purchase the fee interest in private forestlands. Increasingly, a less expensive mechanism is for government or private conservation groups to purchase development rights to private forestlands, leaving fee ownership and forest management in private hands. Either way, it is increasingly necessary to determine which forestlands should be conserved.

Haila (1999) encouraged land managers to answer a series of questions in order to prioritize forestland for conservation. These include:

1) What is the smallest fragment size that provides viable forest habitat for wildlife?
2) What is the maximum distance between forest fragments that will allow fragments to be utilized by wildlife as functionally continuous habitat?
3) What forest types are especially important for habitat continuity?
4) How effective are corridors in promoting dispersal of wildlife species across the landscape?
5) Which species of wildlife can be monitored to indicate habitat condition for a set of target species?

Answers to these questions are species dependent, but considering these types of questions can help determine where to invest limited funding for land protection. Extensive, relatively un-fragmented forest landscapes are an obvious priority for land protection, but what about fragmented landscapes that retain varying amounts of forest cover with varying distances between remaining fragments? An argument could be made that forest fragments in landscapes under development that still provide viable wildlife habitat are the highest priority for protection because land values tend to increase more rapidly in developed than in un-developed areas, and remaining forestlands warrant greater protection due to their higher development threat. At the same time, it is important to know when a given landscape has become too highly fragmented to warrant continuing land protection efforts.

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C. Suppression of Fire and Other Ecological Processes

Many important wildlife habitats are influenced by disturbance agents such as floods, fire, frost, ice storms, and other processes. The following is a brief discussion of some of these processes and the biological consequences of alterations.

Fire Suppression

Fire suppression is defined as all activities associated with extinguishing wildland fires. Fire exclusion is defined as the result of prolonged, successful fire suppression and leads to the conversion of fire dependent systems to closed canopy, more mesic forests.

Lightning ignitions of wildfires are currently rare events in Massachusetts. Some historical records document lightning strikes that resulted in large scale fires, but most fires in southern New England are anthropogenic, and have been for thousands of years.

Descriptions of colonial eastern North America mention the extensive use of fire by native peoples (Stewart 2002, Pyne 1982). Native peoples used fire to attract wildlife and enhance wildlife habitat. Fire was used to clear land, to clear forest understory, to increase berry yields, to drive game, to make passage easier. Some of these ancient fires also escaped, just as they do today, and had unintended consequences. The results of native fire practices and fire management as practiced by European settlers resulted in a rich legacy of fire influenced ecosystems and species in the state, but the widespread and frequent use of fire by native peoples or immigrants was soon challenged and curtailed by settlers, and fires have increased in frequency but decreased in area since the 1700s (Pyne 1982) throughout most of the state. Fire suppression was relatively ineffective in preventing fires from influencing the Massachusetts landscape until the middle of the 20th century, but since the 1960s advances in fire detection and suppression have resulted in fire exclusion from habitats that require fire for their continuing existence.

The conditions created by periodic fire generate habitat for dozens of species of conservation concern in Massachusetts. Of the approximately 115 terrestrial species targeted by this plan, 64 (55%) benefit from conditions created by fire. The range of wildlife species that benefit from increased fire management includes game species such as black bear, wild turkey (Koslowski and Ahlgren 1974) and many terrestrial vertebrates (Wright and Bailey 1982), federally protected species such as the bog turtle, and most of the terrestrial invertebrates targeted in this plan.

Of the 22 macrohabitat types described in this plan, at least nine (41%) are influenced by periodic fire. The beneficial conditions created by periodic fires include the maintenance and restoration of primary breeding, feeding and foraging habitat for at-risk animal species. Pitch pine/scrub oak barrens, young forests, grasslands and rock cliffs are among the many habitats that are perpetuated or enhanced by periodic fire.

One result of fire exclusion is the loss of landscape and habitat heterogeneity, as tree species such as red maples that are fire intolerant come to dominate forests at the expense of oaks which formerly dominated our woodlands (Abrams 1998).

Frequent fires reduce duff layers and remove leaf litter, allowing grasses and forbs to germinate. Grasses and forbs are sources of food and nectar and are vital to many invertebrates targeted by this plan.

Fire exclusion in fire dependent systems such as Pitch Pine/Scrub Oak was intended to reduce the threat to public health and safety posed by these highly volatile fuels. Instead, the exact opposite was achieved and many of these areas are now more dangerous than ever before due to excessive fuel loading. These conditions are also not conducive to the continuing health of wildlife populations dependent on these habitats, as was spectacularly displayed when the habitat required by the extinct heath hen (*Tympanuchus cupido cupido*) was deprived of fire. The last surviving population of this denizen of frequently burned scrub and tree oak barrens was greatly reduced when fuel loads supported unsurvivable fires (Gross 1928, Thompson and Smith 1970).

High severity fires expose mineral soils and kill most trees. Although they occur irregularly, their effects have far reaching consequences for animals such as tiger beetles that require mineral soils. High severity fires are often high intensity fires, while all prescribed fires in Massachusetts are low severity fires. It is a challenge to mimic the conditions created by severe fires. Careful exploration of light scarification techniques is a prerequisite for learning how to restore appropriate patches of sparse vegetation mixed with mineral soil.

Hydrological alteration

Agents of hydrological alteration that degrade aquatic and wetland priority habitats targeted by this plan include impoundments by dams and causeways, stream channelization, road run-off, excessive groundwater extraction, the spread of invasive aquatic plants, bank stabilization, erosion control devices, nutrient enrichment and pollution. Of the 24 habitats targeted by this plan, 17 (71%) are subject to degradation by hydrological alteration.

Riverine flow regimes dictate succession, dispersal of species, nutrients and bed load, species establishment, and virtually every factor important to wildlife habitats (Poff et al. 1997, Nilsson and Svedmark 2002).

Most of the streams and rivers in Massachusetts support at least one dam. Dams prevent fish passage and segregate populations of aquatic animals in general. In addition, dams alter sediment loading, transport and deposition. Dams alter both aquatic and riparian habitats (Collier et al 1996, Postel 2003). Dams alter water temperature as well as flow gradients.

Flood control and impoundment management alter the timing, duration, frequency and intensity of flooding and scouring of instream and riparian habitats.

Water diversions for irrigation or interbasin transfers for water supply change water profiles and temperature regimes, influencing developing fish and their prey and the amount of shoreline suitable for germinating seeds (Richter et al. 2003).

The cumulative impacts of invasive species coupled with hydrological regime alterations can lead to the extirpation of species such as Unionid mussels (Bowers and De Szalay 2004).

Groundwater extraction can deprive wetlands of their annual water budget, leading to the shrinking or disappearance of wetlands dependent on groundwater recharge. In Massachusetts such wetlands are represented by coastal plain ponds, seepage swamps, and Atlantic white cedar swamps, especially in areas of high permeability and transmissivity.

Erosion Control
Ocean beaches are notoriously unstable areas, as they represent the intersection of wave and atmospheric energy. Several imperiled species of animals, including the piping plover and northeastern beach tiger beetle, require dynamic beaches as habitat. They are particularly dependent on blow-outs and overwash fans. Erosion control devices such as groins and jetties alter longshore and on-shore drift of sediments. Well meaning people deposit discarded Christmas trees in dune blow-outs, stabilizing them prematurely and depriving species of important, ephemeral habitat.

Other Agents of Ecological Disruption.
Invasive plants can invade frost pockets, preventing radiational cooling and promoting succession of important shrublands into forests. Invasive plants also alter hydrology by increasing evapotranspiration. Invasives act as process alteration agents by changing the fuels and probability of ignition, altering the burning behavior of fire influenced systems.

Gravel extraction, plowing and harrowing of soils interfere with or eliminate below ground mycorrhizal fungi associations with plant roots. Recovery and restoration of such sites are slowed until natural recolonization by mycorrhizae or innoculation of scarified/sterilized soils is completed.

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D. Exotic Invasive Animals

Environments throughout the world have been modified or disturbed by the introduction of exotic organisms, often with unexpected and detrimental results. Among their impacts, exotics may prey upon native species (Smith 1971, Coblentz and Coblentz 1985, Hughes 1986, Savidge 1987, Bradford 1989, Bailey 1993); hybridize with other species (Heusmann 1974, Braithwaite and Miller 1975, Ferguson 1990); compete with native animals or plants (Woods 1993); depredate or defoliate agricultural crops or forests (Forbush and Fernald 1896, De Vos et al. 1956); introduce diseases or parasites (Jenkins and Winkler 1987, Jenkins et al. 1988, Gogan et al. 1990, Soule 1990); and impede or deplete water supplies (Harris 1971, Courtenay 1978, McMahon 1982). Introduced game species have been regarded as innocuous due to the success of the ring-necked pheasant (McAtee 1945, Allen 1956) and brown trout (Courtenay 1978); however, this perception has been critiqued by Grinnell (1925), Hamerstrom and Hamerstrom (1963), Gullion (1965), Weller (1969), and Bolen (1971).

Established exotics are widespread in North America and occur among many taxonomic groups. Among invertebrates in the U.S., 45 species of earthworms in 18 genera belonging to nine of 12 families are of exotic origin (Gates 1954). At least 126 species of exotic fish (46 of which are established) have been taken from the waters of the continental U.S. (Courtenay et al. 1991). At least 92 species of amphibians and reptiles have been introduced to, or transplanted within, the United States (Smith and Kohler 1978), and at least 29 of them have become established (Collins 1990:39-40). At least 120 species of birds have been successfully introduced or transplanted in North and Central America, the West Indies, and Hawaii (Blake 1975). However, only 25 of 385 North American land mammals derive from exotic sources (Jones et al. 1992). Exotic species have been implicated in the extinction of 68% of 43 taxa of North American fishes (Miller et al. 1989) and 45% of 66 globally endangered, island-endemic birds (Johnson and Stattersfield 1990).

The Division of Fisheries and Wildlife (MDFW) is charged with the responsibility of maintaining the diversity and abundance of the state's habitats, wildlife, fishes, and wild plants (Jones et al. 1988) through sound management practices based on biological data. The MDFW's policy document (Div. Fish. Wildl. 1984:12) provides that "No exotic species will be released until it is determined that its biological requirements and impacts are compatible with the environment and existing wildlife populations, and until the Board has approved the release. Accidentally or illegally released exotics shall be retrieved and destroyed if at all possible." (This policy does not, however, pertain to established exotics, such as brown trout and ring-necked pheasant.) Additionally, Executive Order No. 11987 (May 24, 1977), issued by President Carter, provides that federal executive agencies shall restrict the importation of exotic species into natural ecosystems, encourage states and other agencies to do the same, and restrict the use of federal funds for exporting exotic species.

These mandates, together with the demonstrated adverse impacts of many exotic taxa, demand that the MDFW establish a baseline review of past introductions as a frame of reference for evaluating the effects of exotic organisms in Massachusetts. The MDFW's Nongame Advisory Committee initiated such a review. A total of 263 vertebrate taxa, including 57 fish, nine amphibians, 39 reptiles, 119 birds, and 39 mammals are included in a resulting publication (Cardoza et al. 1993) from which this and the preceding two paragraphs have been largely extracted.

Invertebrates have not yet been thoroughly reviewed by the MDFW, but there are many problem species in this broad category. The Asiatic Freshwater Clam, *Corbicula fluminea*, is now present in the state, and the hemlock woolly adelgid (Orwig and Foster 1998) and the beech scale are widespread. These introduced organisms are expected to have major effects on the Commonwealth's biodiversity as they continue to spread.

An example of an introduced species which has already had widespread effects is the tachinid fly *Compsilura concinnata* Meigen (Boettner et al. 2000). *C. concinnata* is a generalist parasitoid introduced from Europe to control the gypsy moth (*Lymantria dispar* L.) and the brown-tail moth (*Euproctis chrysorrhoea* L.) (Webber and Schaffner, 1926). *C. concinnata* attacks more than 180 species of native Lepidoptera (Schaffner and Griswold, 1934; Schaffner, 1959; Arnaud, 1978), produces three to four generations per year, and can reach very high local densities (Williams et al. 1992). In a field experiment conducted by Boettner et al. (2000), none of the 500 cecropia moth caterpillars (*Hyalophora cecropia* L.) released in central Massachusetts survived to pupation; *C. concinnata* accounted for 81% of the mortality. Boettner et al. (2000) suggest that the promethea moth (*Callosamia promethea* (Drury)) and many *Datana* spp. are similarly impacted by *C. concinnata*. Furthermore, it is possible that other extirpations documented in New England (e.g., the regal moth (*Citheronia regalis* Grote) and the pine devil moth (*Citheronia sepulcralis* Grote and Robinson)), as well as regional declines of the imperial moth (*Eacles imperialis* (Drury)) and the wild cherry sphinx (*Sphinx drupiferarum* J.E. Smith), were hastened by *C. concinnata* or other introduced parasitoids. Most of the species that were extirpated or have declining populations have large larvae that mature in mid- to late-summer when large numbers of *C. concinnata* adults emerge from parasitized gypsy moth caterpillars. Reduced mortality from parasitoids in managed, open shrublands as compared to forested habitats may be important for regionally rare Lepidoptera.

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E. Exotic Invasive Plants

Conflicts between invasive and rare species are of great concern to conservation biologists. Analysis of Federal Register data on threats to 958 species listed as Endangered or Threatened in the U.S. (data from Jun. 1991 to Sept. 1996) revealed that invasive species were implicated in the decline of 42% of the species. For 18% they were the major cause, and for 24% a contributing factor (Stein and Flack, eds. 1996). Even more shocking, Wilcove *et al*. (1998), found that competition or predation by alien species affects 49% of imperiled species in the U.S. and that imperiled plants are affected more than imperiled animals (57% vs. 39%). They concluded that alien species rank second in terms of major threats to biodiversity, with direct habitat destruction or degradation being the only category of threat ranking higher.

One unintended consequence of deliberate disturbance of forest ecosystems is the invasion of these habitats by non-native plant species that have naturalized into the surrounding landscape. This problem is particularly acute in the Northeastern United States, an area that is densely settled and possesses, on average, relatively small units of remaining natural landscape. In Massachusetts for instance, 45% of the total vascular plant flora (Sorrie & Somers 1999) is comprised of naturalized species, and of the 1276 naturalized species, about 5% appear on a list of "invasive" species produced by the MDFW (Weatherbee *et al*., 1998). This list of 67 species represents those regarded as especially aggressive or problematic in minimally managed habitats in the Commonwealth. While 45% non-native species is the highest percentage in New England, the other New England states have alien percentages ranging from 24% in Rhode Island to 35% in Connecticut (Mehrhoff 2000), and the New England flora as a whole is 31% alien (Rejmanek

& Randall 1994, based on Seymour 1982). In a study of Natural Heritage records for regionally rare plants in the New England states, Farnsworth (2004) found that 47% of the 81 species studied had one or more invasive species present at one or more of their population locations.

High non-native percentages are also consistent with specific site inventories conducted in towns and parklands in eastern Massachusetts in recent years. For instance, Bertin (2000) reports that 34% of the 988 plant species present in his recent inventory of Worcester are non-native. In Boston's Middlesex Fells, a 988 acre woodland park established and thoroughly inventoried for plants in 1894, a re-census of the flora in 1993 by Drayton and Primack (1996) concluded that exotic species are increasing in the park at an annual rate of 0.18%, or about one new species every five years (there was a loss of 22 exotic species, but a gain of 36 new ones). Investigators are also reporting a simultaneous decline in native species at inventoried sites. Comparing his current flora to that derived from historical specimens of the town, Bertin concluded that there had been a 17% loss in Worcester's native flora. Similarly, Drayton & Primack (1996) reported that native species declined from 83% to 74% in the Middlesex Fells flora over the past century (133 native species presumed extirpated and only 28 new ones observed).

Islands where introduced species have competed with native flora serve as examples of some of the most serious declines of native species ever documented. In Bermuda, the non-indigenous portion of the flora in 1918 was 65% (Rejmanek & Randall 1994 citing Britton) and the rich flora of Hawaii in 1990 was 47% non-indigenous species (Rejmanek & Randall citing Wagner *et al.*). The native flora on Hawaii has suffered: 800 native species are endangered and more than 200 endemic species are believed to be extinct (Vitousek 1988). Penikese Island in Massachusetts has the same percentage of non-native species as Hawaii, 47% (Backus *et al.* 2002). We need to look at continents as nothing more than big islands, and parks, like Middlesex Fells, as islands of semi-natural vegetation amidst a landscape that is largely alien.

In New England, to return ecosystems to early seral stages for the benefit of native plant and animal species favored by these conditions, one runs the risk of exacerbating the spread of non-native species, especially the invasive ones, into or within these systems. This is a major dilemma for land managers attempting to achieve forestry or biodiversity enhancement goals. Many of the region's most notable invasive species are enhanced by disturbance activities related to routine forestry practices or efforts to restore indigenous wildlife and plants through activities such as prescribed burning, brush-hogging, or mowing. Invasive plant species compete with the indigenous ones, changing forest composition. In an Ohio study, for instance, Hutchinson & Vankat (1997) found that Amur honeysuckle (*Lonicera maackii*) reduces native tree regeneration by shading seedlings. Besides simply forming dense stands or thickets, some invasive species can change ecosystem processes such as soil chemistry, hydrology or fire frequency. Japanese barberry (*Berberis thunbergii*) colonies can change soil pH (Kourtev *et al.* 1998); Phragmites (*Phragmites australis*) can change the hydrology of affected wetlands, and Scotch broom (*Cytisus scoparius*) flammability can alter fire behavior in areas where it has invaded (Richburg *et al.* 2001, Mobley 1954).

If it is accepted that invasive plant species can affect forest regeneration and biological diversity in negative ways, the indigenous biological communities of Massachusetts are facing some serious conservation challenges.

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